

Comparing soil organic carbon dynamics in plantation and secondary forest in wet tropics in Puerto Rico

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Abstract

We compared the soil carbon dynamics between a pine plantation and a secondary forest, both of which originated from the same farmland abandoned in 1976 with the same cropping history and soil conditions, in the wet tropics in Puerto Rico from July 1996 to June 1997. We found that the secondary forest accumulated the heavy-fraction organic carbon (HF-OC) measured by the density fractionation technique, more efficiently than the tree plantation did. Although there was no significant difference in total soil organic carbon (SOC) between the plantation ($5.59 \pm 0.09 \text{ kg m}^{-2}$) and the secondary forest ($5.68 \pm 0.16 \text{ kg m}^{-2}$), the proportion of HF-OC carbon to the total SOC was significantly higher in the secondary forest (61%) than in the plantation (45%) ($P < 0.05$). Forest floor mass and aboveground litterfall in the plantation were 168% and 22.8% greater than those in the secondary forest, respectively, while the decomposition rate of leaf litter in the plantation was 23.3% lower than that in the secondary forest. The annual mean soil respirations in the plantation and the secondary forest were 2.32 ± 0.15 and $2.65 \pm 0.18 \text{ g C m}^{-2} \text{ day}^{-1}$, respectively, with a consistently higher rate in the secondary forest than in the plantation throughout the year. Microbial biomass measured by fumigation–incubation method demonstrated a strong seasonal variation in the secondary forest with 804 mg kg^{-1} in the wet season and 460 mg kg^{-1} in the dry season. However, the seasonal change of microbial biomass in the plantation was less significant. Our results suggested that secondary forests could stock more long-term SOC than the plantations in the wet tropics because the naturally generated secondary forest accumulated more HF-OC than the managed plantation.

Keywords: decomposition, litterfall, long-term carbon, microbial biomass, soil carbon, soil respiration

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Introduction

Tropical forests account for 52% of the world's forest coverage (FAO, 2003) and have experienced frequent land-use changes globally (Silver *et al.*, 2000). In addition to their high economic values, tropical forests are also critical to the global carbon cycle because about half of the world's biomass carbon is stocked in these forests and 14% of the world soil carbon is located in the soils of tropical forests (IPCC, 2000). Therefore, a slight change of these carbon pools could have a

significant impact on global carbon cycle. Although tropical forests are important source and sinks of carbon, there is no consistent conclusion on the net carbon effect of tropical forests on carbon sequestration partially because of the complexity of the forest structures and functions, such as species composition, productivity and decomposition, and intensive management practices (Schulze *et al.*, 2000; Harmon, 2001; Clark, 2002).

Short-rotation plantation, referred to as a Kyoto forest, has been recommended in the Kyoto Protocol as an effective measure to reduce atmospheric CO_2 concentration. Reforestation through plantation on abandoned and degraded agricultural lands in the

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tropics has been proposed as an effective carbon management approach (Montagnini & Porras, 1998). However, some studies have challenged the effectiveness of the Kyoto forest. For example, it was reported that the decomposition of the residuals from previous harvesting would keep the young plantation as a net carbon source for two to three decades (Schulze *et al.*, 2000). Law *et al.* (2003) suggested that old-growth forests might transport more carbon to soils as long-term carbon than a Kyoto forest. Schlesinger & Lichter (2001) found that a young plantation contributed little to long-term soil carbon storage in a 25-year-old loblolly pine plantation in Duke Forest in the US.

Secondary forests are extensive in the tropics and account for more than 40% of the tropical forest land and the area covered by secondary forests is continuously increasing throughout the tropical regions because of fast land-use change (Brown & Lugo, 1990; Hughes *et al.*, 1999). Secondary forests provide important ecosystem services, such as erosion prevention, wildlife habitat improvement, biodiversity maintenance, water conservation and watershed protection (Feldpausch *et al.*, 2004). Sustainable use of secondary forests could reduce the human pressure on primary forests and slow down the conversion of primary forests for agricultural use. A few studies (Weaver *et al.*, 1987; Brown, 1998; Hughes *et al.*, 1999; Guo & Gifford, 2002) on global carbon budget and land-use change showed a great potential for carbon sequestration through reforestation and afforestation of tropical agricultural and pasture lands. The success of the management of tropical forests in the future might well depend upon the adequacy of our ecological understanding of secondary forests. Additionally, accurate calculation of carbon budgets at both national and global scales for the tropical forests depends on our capacity to quantify the accumulation of carbon by secondary forests that are dominant in the tropical forests and have experienced intensive deforestation.

Although many efforts have been made to understand the ecological processes of tropical tree plantations and naturally generated secondary forests (Dixon *et al.*, 1994; Binkley & Resh, 1999; Houghton *et al.*, 2000; Paul *et al.*, 2002), soil and forest floor carbon dynamics, such as carbon pool size, turnover rate, litterfall, decomposition, and soil carbon quality, in these forests mostly remain uncertain. This insufficient knowledge leads to current debating about whether short-rotation plantations, secondary forests or old-growth forests are more effective in sequestering atmospheric CO₂ (Richter *et al.*, 1999; Cox *et al.*, 2000; Schulze *et al.*, 2000; Silver *et al.*, 2000; Wirth *et al.*, 2002; Law *et al.*, 2003).

Soil carbon pool, as the major part of the terrestrial carbon reservoir, plays an important role in the global

carbon cycle. Therefore, the study of soil carbon dynamics is critically important to our ability to understand the carbon balance in these forests and their response to future global change (Davidson *et al.*, 2000). More carbon can be stored below ground by increasing the input rate of organic matter, increasing the depth of carbon stock, boosting the carbon density in the soils, and decreasing the carbon turnover rate in soils (Post & Kwon, 2000). Carbon turnover rate varies considerably among forest soils. Differentiating the total soil organic carbon (SOC) into labile carbon (defined as its resident time from months to several years) and recalcitrant carbon (defined as its resident time from decades to thousands of years) could provide more information in understanding the mechanisms controlling the overall turnover rate of SOC pools in forest ecosystems (Sun *et al.*, 2004). However, direct measurements of the labile and recalcitrant SOC pools have encountered many problems and difficulties, and these pools mostly remain conceptual. An alternative approach to differentiate labile and recalcitrant SOC pools might be measuring the carbon in light fraction (LF-OC) and the carbon in heavy fraction (HF-OC) which is more recalcitrant (Compton & Boone, 2000).

Separations based on density flotation generally have been used for two purposes in studies of SOC: to separate animal and plant debris in LF-OC and to separate organo-mineral associations usually in HF-OC (Stevenson & Elliot, 1989). Although the turnover rates of LF-OC and HF-OC pools vary with different ecosystems, the resident time of HF-OC pools might be longer than of LF-OC pools because LF-OC pools are usually linked to macroaggregates while HF-OC pools are involved in silt- or clay-sized organo-mineral complexes which perform higher functions of physical protection for SOC decomposition. For example, Post & Kwon (2000) pointed out that the turnover of LF-OC in agricultural ecosystems has a bulk turnover time from months to a few years, while the HF-OC is stabilized through microaggregation and its turnover time is on the order of decades.

Using a 20-year-old pine plantation and a secondary forest originated from the same abandoned farmland in the wet tropics in Puerto Rico, we compared SOC pool and carbon quality between the secondary forests and the plantation. Our overall aim was to examine soil carbon quantity and quality as a result of different reforestation programs in abandoned agricultural fields. The specific objectives of this study are to examine: (1) How would litter input and decomposition affect SOC pools and soil activity in the secondary forests and the plantation? (2) Would differences in site conditions and litter chemistry affect litter decomposition rate thus result in a difference in SOC pool between

the secondary forest and the plantation? (3) Would the SOC pools differ in total SOC and HF-OC between the secondary forests and the plantations?

Methods

Study sites

The study was conducted on two sites that were within 100 m distance of each other. One site was in a *Pinus caribaea*-dominated plantation and the other is in a secondary forest. Both sites are located at the Guzman sector of the Luquillo Experimental Forest in north-eastern Puerto Rico (18°18'N, 65°50'W). The plantation and the secondary forests were originated from the same abandoned agricultural land with the same cropping system and management/disturbance history (Lugo, 1992). The sites are characterized by wet tropical climate with mean annual precipitation of 3920 mm and mean annual air temperature of 22.3 °C. The temperature was mild and stable with diurnal and seasonal temperature ranges of 3.5 °C (Fig. 1). Precipitation showed a seasonal variation with greater rainfall between August and October than from January to March (Fig. 1). Soils are classified as mixed isothermic tropohumult in both the plantation and the secondary forest. Physical properties of the soils in the secondary forest and the plantation in a depth of 0–10 and 10–25 cm are shown in Table 1. The sites are relatively flat with a slope of <5° and an elevation of about 400 m above sea level.

The tree plantation was established on cropland that was abandoned in 1976 as part of a reforestation program of the United States Forest Service (Lugo, 1992). The secondary forest has naturally developed on the same abandoned cropland since the same year. The plantation is dominated by *Pinus caribaea* with small trees and grass species underneath the canopy. At the time of our study, started in 1996, the average tree height was about 18 m and the average diameter at breast height (DBH; 1.3 m) was 22 cm in the plantation. The tree density in this plantation approximately was 1100 ha⁻¹. The secondary forest is characterized by a sparse overstory and a dense understory with abundant shrubs and grasses. The dominant canopy species in the secondary forest include *Myrcia splendens*, *Miconia prasina* and *Casearia arborea*, and the major understory species include *Casearia sylvestris*, *Miconia mirabilis* and *Tabebuia heterophylla*.

Field measurement and laboratory

At the plantation and secondary forest sites, we established a 0.25 ha plot for field measurements

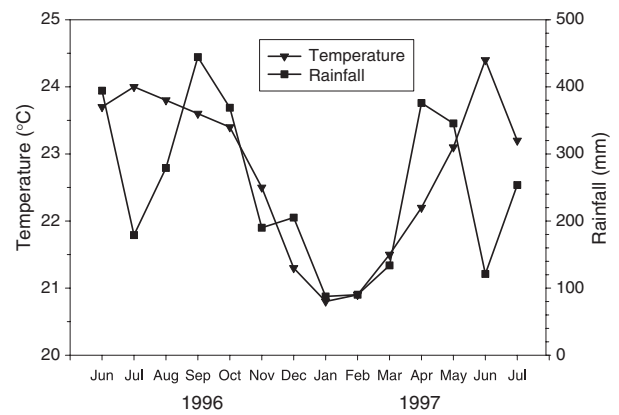


Fig. 1 Monthly precipitation (mm) and air temperature (°C) near the study sites in the Luquillo Experimental Forest in Puerto Rico from June 1996 to July 1997.

Table 1 Soil pH, gravimetric moisture and bulk density in the plantation and the secondary forest

	pH	Soil moisture (%)	Bulk density (g cm ⁻³)
0–10 cm			
Plantation	4.5 (0.05) b	69 (3.4) a	0.61 (0.04) a
Secondary forest	5.1 (0.05) a	61 (7.2) a	0.55 (0.04) b
10–25 cm			
Plantation	4.7 (0.03) b	64 (2.8) a	0.72 (0.05) a
Secondary forest	5.2 (0.06) a	62 (6.5) a	0.58 (0.11) a

Common letters within a column indicate no significant difference between the treatments according to the Tukey's test at $\alpha = 0.05$. Numbers in parentheses are standard error ($n = 4$).

including litterfall, forest floor mass, the rate of litter decomposition, and soil respiration. Litterfall samples in both the plantation and the secondary forest plots were collected every 2 weeks from July 1996 to June 1997 using five randomly located rectangular baskets (0.5 m² in surface area, 0.65 m above ground) lined with 1 mm mesh openings, and sorted into leaves, wood and miscellaneous fractions. Forest floor mass was measured on five 0.5 m × 0.5 m subplots randomly located in both the plantation and the secondary forest every two months from July 1996 to June 1997. Forest floor mass was sorted into the same categories as above-ground litterfall. All plant litter samples were dried to constant mass at 70 °C for determination of mass and chemical analysis. Soil samples (0–10, 10–25 cm) for carbon and soil physical properties were collected from four cores (18.9 mm diameter) in each treatment of the plantation and the secondary forest. These soil samples were air-dried for the laboratory analysis.

Leaf decomposition rate was determined using the mixed-species litterbag method (Berg *et al.*, 1993). Recently fallen leaves and needles were collected from the plantation and the secondary forest. They were placed in litterbags constructed of nylon cloth with 1 mm mesh openings. Each litterbag had a dimension of 25 cm × 25 cm. We placed a total of 160 bags in the secondary forest and the plantation in July 1996. Forty litterbags containing 5 g of oven-dry leaf litter from the pine plantation and 40 litterbags containing 5 g of oven-dry leaf litter from the secondary forest were placed on the forest floors in both the plantation and the secondary forest. These litterbags were collected and weighed after 0, 1, 2, 4, 6, 8, 10 and 12 months. Five litterbags were collected each time at each site. Initial leaf was oven-dried to a constant mass at 60 °C and then ground with a Wiley mill (Thomas Scientific, Swedesboro, NJ, USA) to pass through a 0.85 mm mesh sieve. Total carbon and nitrogen were determined by combustion in a LECO C–H–N elemental analyzer. Elemental concentrations of P, S, Na, K, Ca, Mg, Mn, Fe, and Al were analyzed using a Thermal Jarell Ash Inductively Coupled Plasma Analyzer (ICP) (Scientific Instrument Services, Inc., Ringoes, NJ, USA) after samples were digested with H₂O₂ and concentrated HNO₃ (Liu & Zou, 2002). Initial leaf chemistry in both the secondary forest and the plantation are shown in Table 2. Total C in soils was obtained using a Perkin-Elmer CHN analyzer (Perkin-Elmer Life & Analytical Sciences, Inc., Boston, MA, USA). Soil pH was measured in a 1:1 ratio of water to soil fresh weight with a combination electrode. Soil moisture contents were determined by oven-drying 10 g of randomly sampled fresh soil at 105 °C for 48 h.

Soil respiration (including the litter respiration in this study) was measured using the alkali trap method (Carter, 1993). A plastic chamber with an opening area of 102.5 cm², and a height of 20 cm, and a plastic cup with a diameter of 5 cm, and a height 7.5 cm were used for soil respiration measurements at each location. At each measurement location, a trap unit was prepared by pipetting 15 mL 1.0 M NaOH solution into a plastic cup and placing it on the soil surface. A chamber was immediately placed over the alkali cup and its edge was pressed into soils for 2 cm to ensure the chamber was well sealed. Another cup was filled with the same solution and tightly sealed and placed outside of the chamber as the control to consider the CO₂ absorption during the solution transport. After 24 h, the cup was removed, enclosed with a lid, and taken to the laboratory for analysis. In the laboratory, alkali solutions were titrated with 1 M HCl solution to the phenolphthalein end point to determine the amount of NaOH left after excess BaCl₂ was added to the NaOH

solution to precipitate the carbonate as insoluble BaCO₃. Soil respiration was measured about every 2 months from August 1996 to June 1997.

Soil microbial biomass carbon was measured using the fumigation–incubation procedure of Jenkinson & Powlson (1975) with slight modifications. The flush of CO₂ was determined by titrating with 1 M HCl after 10 days incubation (Liu & Zou, 2002). Soil microbial biomass was measured during a wet season (September 1996) and a dry season (March 1997). The LF-OC was determined using the density fractionation method of Sollins *et al.* (1984). Air-dried soils were passed through a 2 mm mesh sieve and then 1 g of the sieved soil was suspended in 20 mL of NaI solution adjusted to a density of 1.85 g mL⁻¹. The suspension was then sonicated for 15 min at a medium energy level, vacuumed (70 kPa) for 10 min, and then left to settle over night at room temperature to separate light and heavy fractions. The LF-OC at the surface of the density liquid was aspirated and trapped onto a glass fiber filter paper (GF/A), rinsed with deionized water, and then analyzed for total SOC. The HF-OC was determined by subtracting LF-OC from the total SOC.

Data analysis

Soil respiration rate was calculated using the following formula: $C \text{ or } CO_2 = (B - V) / NE$ (Carter, 1993). Decay constant (k), the average rate of litter loss, was determined using the data of litterbag decomposition experiment by the formula: $M_t = M_0 e^{-kt}$ (Olson, 1963). The bulk densities were calculated using the formula: bulk density = (dry weight of sample in grams) / (volume of the sample in cubic centimeters). Significant differences among means of treatments were determined by Tukey's test at $\alpha = 0.05$. Significant differences of seasonal variation of soil respiration, litterfall, and forest floor mass and litter decomposition was tested by Repeated Measures Analysis of Variance.

Results

SOC pool and microbial biomass

Total SOC did not significantly differ between the secondary forest and the plantation. The difference was neither significant in the top 0–10 cm soil layers nor in the 10–25 cm soil layers (Table 3). However, there was a significant difference in HF-OC in both the 0–10 and 10–25 cm soil layer between the secondary forest and the plantation, with higher values in the secondary forest. Microbial biomass was significantly lower in the pine plantation than in the secondary forest in the wet season, but not in the dry season (Fig. 2). In the

Table 2 Initial leaf chemical content in the leaf litter decomposition experiment in the plantation and the secondary forest in Puerto Rico

	Ca (mg g ⁻¹)	P (mg g ⁻¹)	Al (mg g ⁻¹)	Fe (mg g ⁻¹)	Mn (mg g ⁻¹)
Plantation	10.21 (2.01) a	0.29 (0.04) a	1.82 (0.35) b	1.16 (0.22) b	1.09 (0.08) a
Secondary forest	6.31 (0.54) b	0.27 (0.11) a	6.12 (0.16) a	3.65 (0.37) a	0.82 (0.07) a

	K (mg g ⁻¹)	Mg (mg g ⁻¹)	C (%)	S (%)	N (%)
Plantation	8.84 (0.49) a	2.24 (0.25) a	49.7 (6.6) a	0.79 (0.04) a	1.22 (0.06) b
Secondary forest	1.98 (0.24) b	1.94 (0.13) a	51.8 (5.8) a	0.17 (0.02) b	1.88 (0.08) a

Common letters within each column indicate no significant difference by the Tukey's test at $\alpha = 0.05$. Numbers in parentheses are standard error ($n = 4$).

Table 3 SOC in the plantation and the secondary forest

	HF-OC (kg m ⁻²)	Total SOC (kg m ⁻²)
0–10 cm		
Plantation	1.16 (0.08) b	3.27 (0.08) a
Secondary forest	1.84 (0.12) a	3.45 (0.11) a
10–25 cm		
Plantation	0.98 (0.09) b	2.32 (0.14) a
Secondary forest	1.26 (0.07) a	2.23 (0.09) a

HF-OC, heavy fraction organic carbon; SOC, soil organic carbon.

Common letters within a column indicate no significant difference between the treatments according to the Tukey's test at $\alpha = 0.05$. Numbers in the parentheses are standard error ($n = 4$).

secondary forest, microbial biomass was 804 mg C kg⁻¹ in the wet season and 460 mg C kg⁻¹ in the dry season. While the corresponding values in the plantation were 550 and 480 mg C kg⁻¹, respectively.

Forest floor mass

Total forest floor mass (i.e. sum of leaf litter, wood and miscellaneous litter, mean \pm SE) was significantly ($P < 0.05$) higher in the plantation (1572 ± 48 g m⁻²) than in the secondary forest (587 ± 14 g m⁻²) (Fig. 3). Leaf litter and wood were significantly higher in the plantation, while the miscellaneous litter (mainly composing of fruits, flowers and semi-decomposed scrapes) was significantly higher in the secondary forest. Leaf litter, wood and miscellaneous litter in the plantation accounted for 74%, 13% and 3% of the total forest floor mass, respectively. The proportion of leaf litter, wood litter and miscellaneous litter to the total forest floor mass in the secondary forest was 81%, 3% and 16%, respectively. Total forest floor mass, leaf, wood, miscellaneous litter did not differ between the

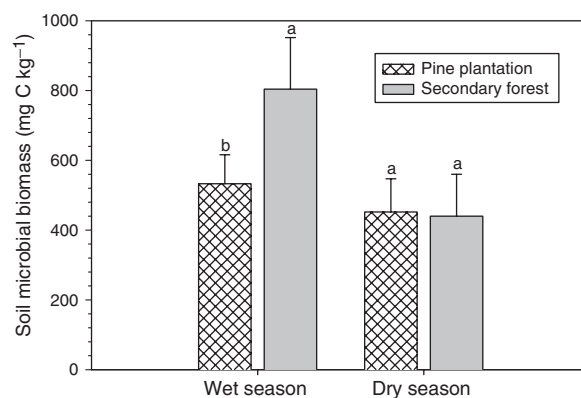


Fig. 2 Mean (\pm standard error) of microbial biomass (mg C kg⁻¹ soil) of the secondary forest and the plantation in the wet (August) and dry (March) seasons. Common letters within a season indicate no significant difference between the treatments 95% confidence level. $n = 4$.

dry season (March) and the wet season (September) in both the plantation and the secondary forest.

Soil respiration

Total soil respiration (including litter respiration, mean SE) was significantly higher in the secondary forest (2.65 ± 0.18 g C m⁻² day⁻¹) than in the plantation (2.32 ± 0.15 g C m⁻² day⁻¹). Seasonal patterns of soil respiration were found in both the plantation and the secondary forest (Fig. 4). Soil respiration at both sites reached the lowest level ($P < 0.05$) in April 1997, immediately following the dry period in March in Puerto Rico.

Litterfall

Mean total litterfall rate (mean \pm SE) was significantly higher in the plantation (2.61 ± 0.09 g m⁻² day⁻¹) than

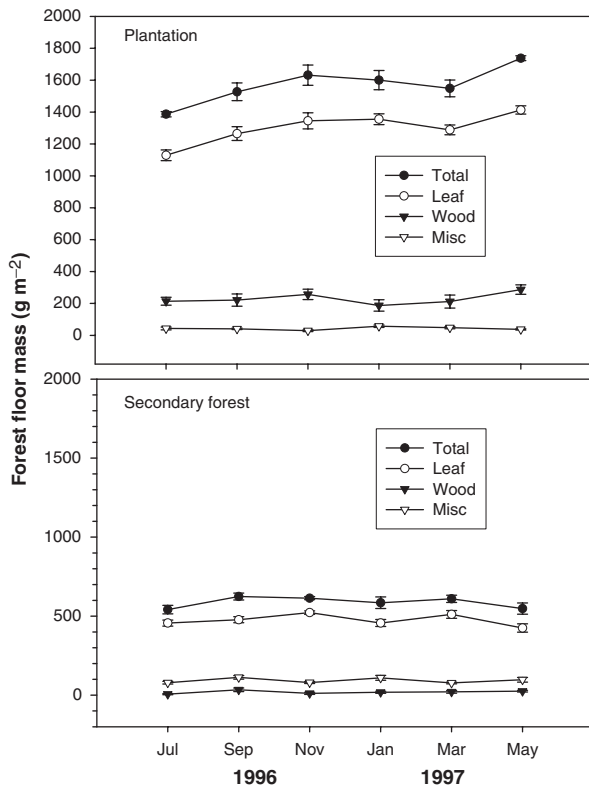


Fig. 3 Monthly forest floor mass (mean \pm standard error) of the secondary forest and in the plantation from July 1996 to May 1997. $n = 4$.

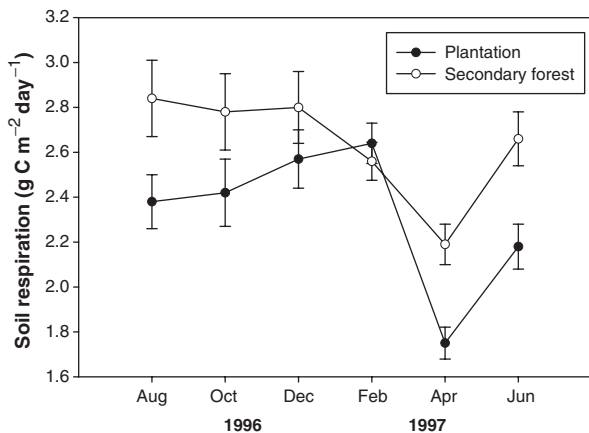


Fig. 4 Mean (\pm standard error) of soil respiration ($\text{g C m}^{-2} \text{ day}^{-1}$) and the seasonal variation in the secondary forest and the plantation from August 1996 to June 1997. $n = 4$.

in the secondary forest ($2.14 \pm 0.04 \text{ g m}^{-2} \text{ day}^{-1}$) (Fig. 5). Total annual litterfall in the secondary forest and the plantation was 781 and 959 g m^{-2} , respectively. Leaf, wood, and miscellaneous litter accounted for 80%, 3% and 17% of the total litterfall in the secondary forest, respectively. In the plantation, the corresponding values were 77%, 18% and 5% of the total litterfall,

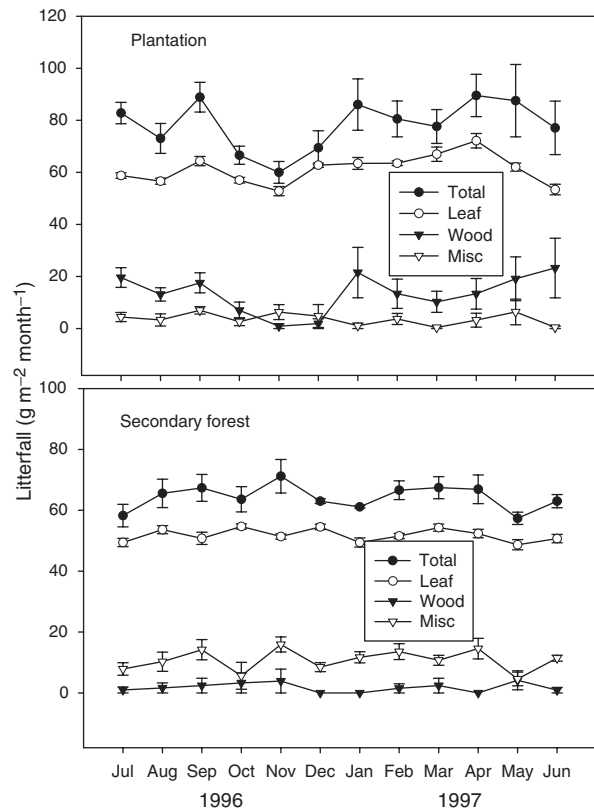


Fig. 5 Mean monthly litterfall (mean \pm standard error, $n = 4$) in the secondary forest and in the plantation from July 1996 to June 1997.

respectively. The rate of total litterfall was significantly lower in December 1997 but there was no significant difference of the rate of total litterfall between the in the dry season (March) and in the wet season (September) in the plantation. Leaf and wood litterfall had similar patterns with the total litterfall in the plantation. We did not find significant difference between the dry season and the wet season in total, leaf, wood and miscellaneous litterfall in the secondary forest (Fig. 5).

Leaf litter decomposition

Leaf decomposition rate was not significantly different between the plantation and the secondary forest given the leaf litter of the same species composition (Table 4, Fig. 6). However, under the natural conditions (the secondary forest leaves were incubated in the secondary forest and the plantation leaves were incubated in the plantation), the leaf litter in secondary forest decomposed significantly faster than the leaf litter in the plantation. The mass loss rate of plantation leaves incubated in the plantation ($72\% \text{ yr}^{-1}$, in comparison with its initial mass) and in the secondary forest ($68\% \text{ yr}^{-1}$) were significantly lower than the secondary forest

Table 4 Mean (standard error) annual decay constant, k , in the plantation and the secondary forest

Leaf source	Incubation site	k	R^2	P	Mass loss% yr ⁻¹
Plantation	Plantation	1.13 (0.23)	0.58	<0.03	71.7 (0.72) b
Plantation	Secondary forest	0.94 (0.11)	0.64	<0.01	67.6 (0.68) b
Secondary forest	Plantation	1.48 (0.17)	0.69	<0.00	79.4 (0.96) a
Secondary forest	Secondary forest	1.45 (0.09)	0.73	<0.00	78.2 (0.58) a

Values in the parentheses were derived from six-time sampling with five replicates. Common letters in each column indicate no significant difference by the Tukey's test at $\alpha = 0.05$.

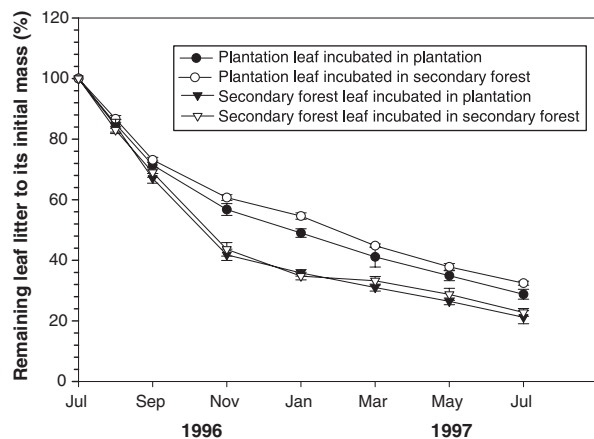


Fig. 6 Leaf litter decomposition (represented by the percentage of leaf litter mass remained in the litterbags) (mean \pm standard error, $n = 5$) in the secondary forest and the plantation from July 1996 to June 1997.

leaves incubated in the plantation (79% yr⁻¹) and in the secondary forest (78% yr⁻¹), respectively (Table 4). The mass loss rate declined with incubation time. No significant difference in mass loss rate was found during the first 2 months regardless of leaf quality and ecosystem types under this study. The most striking difference of mass loss rate between bags of different leaf litter quality was detected from the third to sixth months of the incubation period. The decay constant, k (mean \pm SE), of pine needles was 1.13 ± 0.23 in the plantation and 0.94 ± 0.11 in the secondary forest. The value of k of the secondary forest leaves was 1.48 ± 0.17 when placed in the plantation and 1.45 ± 0.09 when placed in the secondary forest.

Discussion

The physical fractionation of SOC into LF and HF is an effective approach in differentiating SOC by quality. Physical fractionation of SOC emphasizes the role of soil minerals on its turnover, which relates more directly to SOC dynamics *in situ* than classical wet chemical SOC fraction (Elliot & Cambardella, 1991; Sollins *et al.*, 1996; Swanston *et al.*, 2002). In addition, we

chose the physical fractionation techniques to separate the SOC pools into LF-OC and HF-OC because the technique is simple and widely used (Sollins *et al.*, 1996; Sun *et al.*, 2004). Physical protection, chemical stabilization and biochemical stabilization are three main mechanisms of SOC stabilization (Christensen, 1996). Heavy fraction of SOC can be stabilized in microaggregates which physically protect SOC by establishing physical barriers between microbes and enzymes and their substrates (Elliot & Coleman, 1988), thus HF-OC could have longer resident time in soils than LF-OC usually associated with macroaggregates, though part of LF-OC can also be stabilized in macroaggregates as intra-aggregate particulate carbon (Cambardella & Elliot, 1993; Post & Kwon, 2000). Our results indicate that this differentiation is informative and critical to our understanding of the soil carbon turnover and dynamics in forest ecosystems. For instance, the total SOC stocks were similar in the plantation and the secondary forest but the HF-OC was significantly higher in the secondary forest than in the plantation.

Below-ground carbon storage, especially the long-term carbon storage, is critical to the estimate of effective carbon sequestration capacity of ecosystems, such as forest, cropland and range and grassland, through appropriate management practices. Although increasing attention has been paid to tropical forests, no consistent conclusions have been made on the effects tropical forest soils on carbon sequestration. Richter *et al.* (1994) found that SOC showed little increase in the topsoil layer (0–7.5 cm) after 28 years of loblolly pine growth on previously cultivated land. Bashkin & Binkley (1998) reported that there was no net increase in total SOC after 10–13 years of afforestation in the top 55 cm of soil in Hawaii. On the same site, Binkley & Resh (1999) found no net change of total subsoil carbon after conversion of sugarcane to forest whereas ¹³C measurements showed a fast replacement of the sugarcane carbon with forest-derived carbon. Schlesinger and Lichter (2001) suggested that significant long-term net carbon sequestration in forest soils is unlikely based on a study in a 25-year-old loblolly plantation in Duke Forest in North Carolina using

isotopic analysis. However, some other studies showed very positive conclusions on soil carbon sequestration. Silver *et al.* (2000) found that soil carbon accumulated at a rate of $1.36 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ during the first 20 years of afforestation in the wet tropics in Puerto Rico. Del Galdo *et al.* (2003) reported that afforestation by 20 years increased total soil carbon by 23% and forest-derived carbon contributed 43% to the total carbon in the top 10 cm of soils in north-eastern Italy. Thuille *et al.* (2000) reported that carbon accumulated at a rate of $0.36 \text{ Mg C ha}^{-1} \text{ year}^{-1}$ in the organic soil layer during the first 62 years of reforestation. Guo & Gifford (2002) reported in a review that soil carbon stocks increase by 53% after land use change from crop to secondary forest and by 18% from crop to plantation. In this study, we did not find significant difference in total SOC between the plantation and the secondary forest after 20-year reforestation from abandoned farmlands. However, significant difference may exist on total SOC between the two ecosystems if soil depth was adjusted to account for the difference in soil bulk density. With only two soil depths, we could not perform some curve fitting for the percentage of C with soil depth, and then adjust the values at the second sampling time. The 11% lower bulk density (0–10 cm) in secondary forest would actually be about 10% more than the value indicated in the table (Table 3), as no adjustment for bulk density changes were made. For the 10–25 cm depth, the bulk density in the secondary forest was 24% lower, so the actual effect of the secondary forest might be on the order of 25% greater than the unadjusted values indicated in the table.

Comparison of different types of forests is an effective way to obtain sound knowledge on the mechanisms controlling SOC budgets. SOC in tropical forest ecosystems comes from net primary production (NPP) mainly through aboveground litterfall and root turnover and its output is mainly through autotrophic and heterotrophic soil respiration and the leaching of dissolved organic carbon. The stock of SOC in forest represents the accumulative difference between the input from NPP and the output through decomposition and leaching. The higher litter decomposition and turnover in the secondary forest could result in more HF-OC accumulating in the secondary forest. The same indication could be reached by the comparison of microbial biomass, which is much higher in the secondary forest in the wet season.

In the present study, we found that the plantation with more forest floor mass had lower HF-OC than the secondary forest which stocked less forest floor mass, suggesting that litter quality plays an important role in the regulation of relative proportion of LF-OC and HF-OC. The higher accumulation rate of forest floor mass

in the plantation than in the secondary forest is because of the higher input rate of litterfall and the lower decomposition rate in the plantation. The higher litterfall rate in the plantation might be caused by the higher productivity in the plantation (Lugo, 1992) and the slower decomposition in the plantation might relate to litter quality (e.g. lower nitrogen concentration in leaf litter in the plantation) (Table 2). Tanner (1991) reported that leaf decomposition rate was positively correlated with leaf nitrogen content in a decomposition experiment (15 species studied) in tropical forests in Jamaica. Additionally, the higher productivity in the plantation might enhance the competition between plants and soil microbes for nitrogen, especially in the tropics where the ecosystems are nitrogen limited (Hu *et al.*, 2001). However, our result that the leaf litter from the plantation decayed more slowly than that from the secondary forest in both sites in our litter bag reciprocal experiment suggests that litter quality, instead of soil physical and biological factors, might dominate the decomposition processes in these forests.

Soil respiration measured at soil surface including microbial decomposition and root respiration provides critical knowledge on understanding soil carbon budgets and dynamics. Our result that the plantation had a lower respiration rate than the secondary forest indicated that the decomposition was slower in the plantation than in the secondary forest. This conclusion may be compromised by considering root respiration because the secondary forest has a larger amount of root biomass than the plantation (Lugo, 1992) and root respiration may account for 30–90% of the total soil respiration (Bowden *et al.*, 1993; Thierron & Laudelout, 1996; Epron *et al.*, 1999; Xu & Qi, 2001). In addition, the alkali-trap method we used in this study might underestimate soil respiration in comparison with the infrared gas analyzer technique (Jensen *et al.*, 1996; Yim *et al.*, 2002). We do not expect these biases will change our conclusion on the comparison of soil respiration between the plantation and the secondary forest.

The percentage of 1-year weight loss from the plantation leaf litter (71.7% in plantation site, 67.6% in the secondary forest) in our study was much higher than the litter loss (14–17%) in a Mediterranean pine forest reported by Moro & Domingo (2000). The decay constant k of the plantation leaf litter (1.13 in the plantation site and 0.98 in the secondary forest site) was higher than the k -value (0.48) reported by Berg *et al.* (1993) in a pine forest of Blackhawk Island. The higher decay rate of the leaf litter in the pine plantation in the present study may be explained by the fact that leaf material used for this decomposition experiment was a mixture of pine needles and broadleaves, while the

materials used for their decomposition studies were purely needles. In addition, warmer weather conditions could also contribute to the faster decomposition in our study site. However, the percentage of the 1-year weight loss in this study was within the range from 27% to >96% in a decomposition study for freshly fallen leaves of 15 species in Jamaica tropical montane forests (Tanner, 1991).

In conclusion, the effects of plantations and natural secondary succession generated from abandoned agricultural lands on soil carbon stocks might vary through in carbon quality, in terms of HF-OC proportion to total SOC, during the land-use change. Our results provide first-hand information for the evaluation of tropical plantations and secondary forests in terms of the effectiveness of soil carbon sequestration.

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